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BREEDING HABITAT AND LOCAL POPULATION SIZE OF THE OREGON SPOTTED FROG (*RANA PRETIOSA*) IN OREGON, USA

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ABSTRACT—Distribution of the Oregon Spotted Frog (*Rana pretiosa*) has shrunk markedly, and more than two-thirds of known populations are located along the Cascade Range in central Oregon. Despite conservation concern, little is known about how habitat attributes and stressors such as invasive species influence *R. pretiosa* populations. We used egg mass counts to study *R. pretiosa* habitat relationships at oviposition sites and breeding ponds. Oviposition sites were in shallow water above gradually sloping substrates that supported moderate or dense herbaceous vegetation. Sixty-one per cent of occupied breeding ponds had fewer than 20 egg masses. We found strong support for 2 predictors of egg mass count: positive effect of other *R. pretiosa* breeding sites nearby, and a negative effect of non-native fish having access to preferred *R. pretiosa* overwintering habitat. We found moderate support for effects of emergent and submergent vegetation coverage (positive), and of ponds being located in the Klamath River Basin (negative). Maintaining and restoring overwintering habitats that are free of non-native game fish is likely to benefit *R. pretiosa* in our study area. Further work on movement ecology is needed to improve our understanding of habitat connectivity and the effects of site isolation on the persistence of *R. pretiosa* in Oregon.

Key words: Oregon Spotted Frog, *Rana pretiosa*, habitat, non-native trout, wetland, Deschutes, Klamath

The Oregon Spotted Frog (*Rana pretiosa*) is estimated to be extirpated from >70% of its range in the Pacific Northwest, which historically extended from northeastern California to southwestern British Columbia (McAllister and others 1993; Hayes 1997). Recent assessments have identified approximately 33 localities that are likely separate populations in British Columbia (3) and the states of Washington (6) and Oregon (24) (Hayes 1997; Watson and others 2003; Cushman and Pearl 2007). *Rana pretiosa* is considered extirpated from the Willamette Valley, northeastern California, and much of its range in western Washington (Hayes 1997; Pearl and Hayes 2005). Loss and modification of wetland habitat are hypothesized as leading causes of declines, and effects of habitat alterations are likely exacerbated by other stressors such as introduced fish and American Bullfrogs (*Lithobates catesbeianus*) (Hayes and Jennings 1986; Pearl and others 2004). More

than two-thirds of known *R. pretiosa* populations range-wide and all populations in Oregon are found along the crest and eastern slope of the Cascade Range. Most of these are at least partly on public land. Management options are limited by a lack of information on the relative importance of habitat attributes and stressors that might affect population size or persistence.

Several aspects of *R. pretiosa* life history and habitat use make it a species with high vulnerability to changes in wetland habitat and predator communities. In our study area, *R. pretiosa* breed explosively soon after wetlands thaw, and are often finished within 1 to 2 wk (Pearl and Hayes 2005). Females are thought to breed only once per year and lay 1 egg mass per breeding bout, and egg masses are relatively large and easily detected (Licht 1971; Nussbaum and others 1983). Egg masses are usually deposited communally in a small number of oviposition sites in shallow water (Licht 1971).

Larvae transform in mid- to late summer and are not known to overwinter (Nussbaum and others 1983). Post-metamorphic stages make extensive use of vegetated shallows in summer, when most growth occurs (Watson and others 2003; Chelgren and others 2008). Oregon Spotted Frogs rely on habitats that do not freeze, such as springs, flowing channels, or deep open water during the long (4 to 6 mo) winters in our study area (Hallock and Pearson 2001; Shovlain 2005; Chelgren and others 2008). Movement among these seasonally important microhabitats appears to be strongly associated with flooded or saturated corridors (Watson and others 2003).

In this study we used a census of *R. pretiosa* egg masses to provide: 1) an improved description of microhabitats used for breeding; 2) an updated assessment of the status of the species in Oregon; and 3) an evaluation of habitat and stressor variables that have potential to influence breeding numbers at the scale of breeding ponds. We focused on a subset of habitat and stressor variables that have been hypothesized to affect *R. pretiosa* or other western ranid frogs. For example, non-native fish are widespread and implicated in reducing populations of several ranids, including the closely related Columbia Spotted Frog (*Rana luteiventris*) (Hayes and Jennings 1986; Monello and Wright 1999; Knapp and Matthews 2000). Less is known about which life stages are most affected by fish or whether these effects can be mediated by the amount of access fish have to specialized habitats used by *R. pretiosa* for breeding or overwintering (for example, see Pilliod and Peterson 2001). Our aim was to provide information relevant for conservation of *R. pretiosa* populations, as well as for prioritizing surveys at sites that have potential to harbor new populations.

METHODS

The study area flanked the Cascade Range in Oregon, USA. Survey sites were mainly in the upper Deschutes and upper Klamath River basins on the east slope of the Cascade Range, with a small number of sites west of the Cascade crest in the Willamette and Sandy river basins. Elevations of study sites ranged from 957 to 1612 m ASL. The western part of the study area (along the Cascade Range crest) receives more moisture than the eastern portion of the study area. Vegetation is primarily closed

forest west of the crest and grades to open, drier forest and shrubland in the southeastern portion of the range. Forests in the west are dominated by Mountain Hemlock (*Tsuga heterophylla*), Lodgepole Pine (*Pinus contorta*), and Douglas-fir (*Pseudotsuga menziesii*); and in the east by Lodgepole and Ponderosa Pine (*Pinus ponderosa*) (Franklin and Dyness 1977).

Site Selection

We sought to survey all known and suspected *R. pretiosa* breeding sites in the core of the species' extant range in Oregon (Fig. 1). We did not include the Willamette Valley in our study because previous surveys suggest the species is extirpated from the region (Hayes 1994). The sample pool mainly comprised potential breeding ponds in and around wetland complexes where we have encountered *R. pretiosa* in our past work. We also included a small number of sites where: 1) *R. pretiosa* have been reported by biologists familiar with the species; or 2) museum collections indicated the species formerly was found but where previous surveys had not detected them (Hayes 1994; Cushman and Pearl 2007; R Demmer, M Parker, pers. comm.). Where land ownership and time allowed, we also surveyed lentic wetlands within 0.5 to 1 km of sites that had records of *R. pretiosa*. We considered potential breeding wetlands to be separate sampling units (hereafter 'breeding ponds') if they were separated by land or significant barriers such that *R. pretiosa* life stages would not be able to move between them without leaving water in any but the highest flood years. Most breeding ponds ($n = 82$) were on public lands managed by the US Forest Service, US Bureau of Land Management, and US Fish and Wildlife Service, but we also included private sites ($n = 23$) where we were granted access by landowners. We were unable to completely survey a few large historic sites (Crane Prairie Reservoir, Davis Lake, and portions of Klamath Marsh National Wildlife Refuge), and these were not included in our analyses.

Data Collection

Counts of egg masses provide a useful index of local breeding population size for species such as *R. pretiosa* that breed explosively and have conspicuous egg masses (Crouch and

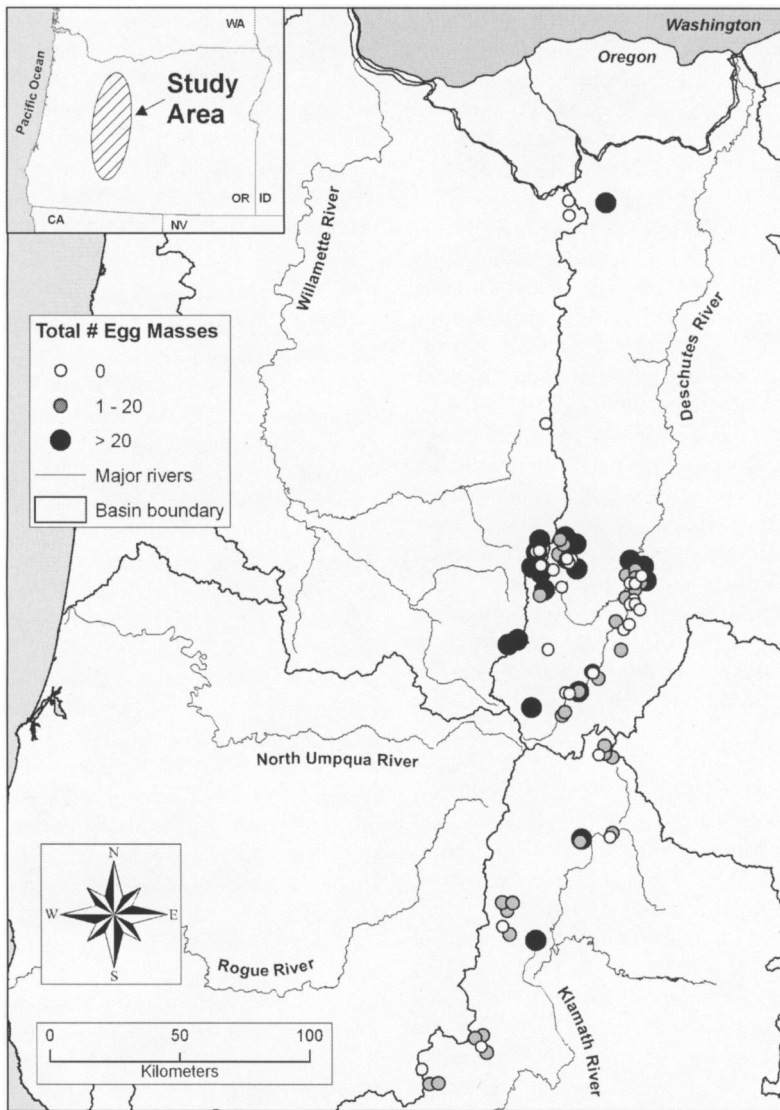


FIGURE 1. Locations and egg mass counts for sites surveyed for *Rana pretiosa* breeding in Oregon ($n = 105$, including Big Marsh).

Paton 2000; Skidds and others 2007). We counted egg masses with survey teams of 1 to 16 observers over 1 to 3 surveys between 13 April and 14 June 2006. We tracked progression of breeding at a subset of sites to time surveys when breeding had concluded and prior to hatching. We took GPS location data at oviposition sites and marked them on site maps to help identify new egg masses on subsequent surveys. We used the total number of egg masses as an index of abundance.

Oviposition sites.—We defined an oviposition site as the center of a cluster of egg masses when laid in aggregations (the typical arrangement) or the center of an egg mass if it was laid >20 cm from any other egg mass. We assessed variables upon our 1st detection of each oviposition site, and the methods followed Pearl and others (2007). Variables assessed were: water depth (cm); substrate slope (%); distance from shore (m); dominant vegetation (sedge/rush, grass, cattail, moss, algae, shrub, or open);

vegetation density (0; 1 to 33%; 34 to 66%; or >66%); angle from pond center to oviposition site (compass bearing 0 to 359°, where 0° represents an oviposition site that is due north of pond center); and shading (average inclination of tree line or other sunlight obstructions measured at 7 evenly spaced points along the southern horizon). We tried to collect data at all oviposition sites in all breeding ponds, but were constrained by time. We present summary statistics (mean \pm SE; median, range) on attributes of oviposition sites (S-PLUS 6.2, Insightful Corp., 2003). Angle from pond center is presented as mean and 95% confidence interval (Oriana 2.0, Kovach Computing Services, Inc.).

Breeding ponds.—We used spring breeding surveys and late summer revisits (August through October 2006) to assess habitat and biotic attributes that have been proposed as affecting *R. pretiosa* or its sister species, *R. luteiventris* (Nussbaum and others 1983; Hayes 1997; Pilliod and Peterson 2001; Pilliod and others 2002; Cushman and Pearl 2007). We categorized the general habitat type for each potential breeding site as: marsh; pond-lake; beaver dam; oxbow; or constructed (anthropogenic pond, reservoir, or ditch). We scored maximum depth as <1, 1 to 2, or >2 m, by visual inspection or measurement with a 2-m stick. During our spring surveys, we visually estimated the proportion of each pond in emergent and submergent vegetation. We multiplied this proportion by the total pond area (derived from post-1990 aerial photography) to generate an estimate of total area (m²) in emergent and submergent vegetation. We used a $\ln[x + 1]$ transformation for the emergent-submergent vegetation cover values to stabilize the variance.

We scored a variable quantifying potential overwintering habitats available to *R. pretiosa* by counting springs, flowing channels, and basins of open, deep water that were linked by surface water with each breeding pond. This is reasonable since *R. pretiosa* mainly move via aquatic or saturated routes, and rarely make longer movements across terrestrial habitats (Hallock and Pearson 2001; Watson and others 2003).

We scored a binomial variable (hereafter 'breeding nearby') to indicate whether any other *R. pretiosa* breeding pond is known to exist within typical movement distance for the

species (≤ 0.5 km if ponds were separated by upland; ≤ 2.0 km if connected by a riparian corridor). These distances are derived from telemetry and mark-recapture data from 4 sites in Oregon (Shovlain 2005; Chelgren and others 2008; J Bowerman, C Pearl, unpubl. data).

We assessed several biotic attributes in addition to the structural habitat variables. We categorized American Beaver (*Castor canadensis*) activity as active, historical (no fresh evidence), or not detected based on our observation of beavers or their sign (dams, lodges, chewed trees or branches). Based on our observations of livestock footprints, feces, and browse, we categorized grazing status for each site as evidence in >50% of site, evidence in <50% of site, or not detected. We also noted detections of invasive American Bullfrogs (present or not detected) and Reed Canary Grass, *Phalaris arundinacea* (visual estimation of percent cover).

If predatory fish have access to habitats where *R. pretiosa* life stages are vulnerable, such as areas used by young tadpoles or where post-metamorphic stages spend winters, there is potential for fish to reduce survival and population size. This scenario has been suggested by Pilliod and Peterson (2001) with overwintering Columbia Spotted Frogs in montane sites in Idaho. We scored a binomial variable for non-native fish access to *R. pretiosa* overwintering habitat (as described above): 0 = non-native game fish either not known from site or have access to <50% of the overwintering habitat we identified; 1 = non-native game fish have access to >50% of potential overwintering habitat. We used a 3-level variable to describe non-native fish access to breeding and larval rearing habitat: 0 = non-native game fish not known to occur at site; 1 = non-native game fish present, but their access to breeding habitat is limited by logs, dense vegetation, substrate topography, or distance from fish habitat; 2 = non-native game fish have direct access to habitats. To score this variable, we considered breeding habitat to be the areas where we found egg masses, or areas that met the criteria described by our oviposition site data. The non-native game fish included in this variable are known or suspected predators of amphibians (Kiesecker and Blaustein 1998; Monello and Wright 1999; Pilliod and Peterson 2001); primarily Brook Trout (*Salvelinus fontinalis*), but also Brown Trout (*Salmo trutta*), Rainbow Trout

(*Oncorhynchus mykiss*), Atlantic salmon (*Salmo salar*), and centrarchids (*Micropterus* and *Lepomis* spp.). We did not include bullheads (*Ameiurus* spp.) or small fish such as sticklebacks (*Gasterosteus* spp.) that are specialist feeders on invertebrates because we found no information on their predation on amphibians, and because predation by these taxa on *R. pretiosa* is likely to be limited due to fish size, diet, or habitat affinities.

Analysis of Factors Affecting Egg Mass Counts in Breeding Ponds

Data from 104 potential breeding ponds was used in our analysis of predictors of *R. pretiosa* breeding abundance. We excluded data for 1 site, Big Marsh (1733 egg masses [J Kittrell, Deschutes National Forest, pers. comm.]; Deschutes River basin), from the analysis because the total habitat area was more than 7 times the next biggest site, and its egg mass count was more than 3 times as large as the 2nd highest count. Thus, our scope of inference for the modeling applies to potential breeding sites other than Big Marsh. This should not affect the interpretation of results in Oregon, since there are no known breeding sites that approach the size of Big Marsh.

For analyses of all potential breeding ponds except Jack Creek, our response variable was the total number of *R. pretiosa* egg masses counted in spring 2006. The Jack Creek population has declined steadily since monitoring began in 1999, from 335 egg masses to about 5% of initial breeding (17 egg masses in 2006; T Forbes, L Peterson, T Simpson, J Oertley, Winema National Forest, unpubl. data). Declines at Jack Creek have not followed obvious changes in habitat conditions or stressors, and contrast with data from other *R. pretiosa* sites in the study region for which parallel breeding census data are available ($n = 5$ populations). Because the other monitored populations appear to be stable or increasing over the same time period and we are not confident that our predictor variables can account for the decline, we elected to use an average egg mass count (1999 through 2006) for those breeding sites in Jack Creek rather than the 2006 count. This decision effectively treats Jack Creek as a "good" *R. pretiosa* habitat despite the observed decline, and is consistent with our objective to compare habitat quality among sites. It assumes

that the recent declines are not related to the habitat variables we assessed.

We used multiple regression and an information-theoretic approach to rank predictors of abundance of *R. pretiosa* egg masses (Akaike 1985; Burnham and Anderson 2002). We screened predictors for collinearity, but found no r values ≥ 0.7 , so retained the whole group (Bradford and others 2003). We modeled the abundance of egg masses with a negative binomial distribution, which is appropriate for count data like ours where the variance greatly exceeds the mean (White and Bennetts 1996; Ver Hoef and Boveng 2007). Modeling count data with the negative binomial distribution also relaxes the assumption of spatial independence relative to the Poisson distribution (White and Bennetts 1996). We fit the negative binomial models with the `glm.nb` function from the MASS library for S-PLUS (Venables and Ripley 1999). We chose to estimate the overdispersion parameter (θ) for each model rather than specifying a fixed parameter to use in all of the models. We included Basin (Klamath vs. Deschutes/Willamette) in all models because of soil and climatic differences and the potential for *R. pretiosa* genetic differences between the 2 basins (Funk and others 2008).

Our ultimate goal was to compare the strength of our selected variables for *R. pretiosa* breeding rather than to compare the models. Thus, we calculated a weighted average of the coefficients for each variable based on the model weights. We considered 131 models comprising all combinations of ≤ 3 variables (excluding Basin) as main effects. We ranked models with Akaike's Information Criterion for small samples (AICc) and calculated model weights (w_i), which provide a probability that the model is the best of the candidate models given the data (Burnham and Anderson 2002). The weights of models that contained a given variable were summed to calculate the variable weight, which is the probability that a given variable should be included in the best of the candidate models. The unconditional standard error which takes model selection uncertainty into account for each variable was weighted by the variable weight and summed across all models containing the predictor as per Burnham and Anderson (2002). The unconditional standard error was calculated by summing the

TABLE 1. Characteristics of oviposition sites ($n = 228$) of *Rana pretiosa* in Oregon.

Variable	Mean ($s_{\bar{x}}$)	Median	Range
Depth (cm)	18.5 (0.75)	16.0	1–57
Distance from shore (m)	4.1 (0.3)	2.0	0.08–35.0
Substrate slope (%)	8.8 (0.6)	5.0	0–58.3
Shading on southern horizon (0–90°)	12.5 (0.5)	10.6	1–37.7

following formula for all models that included a given variable:

$$\left\{ \begin{aligned} &(\text{standard error of variable from model})^2 \\ &+ [(\text{model averaged estimate of variable}) \\ &- (\text{variable estimate from model})]^2 \end{aligned} \right\}^{1/2} \times (\text{model weight}).$$

We used the unconditional standard errors to calculate 95% confidence intervals around each odds ratio.

RESULTS

Oviposition Sites

We quantified all habitat variables except angle from pond center at 228 oviposition sites in 46 of the 60 ponds in which we found *R. pretiosa* breeding. Oviposition sites tended to be in shallow water above vegetated, shallowly sloped substrates, and relatively close to shore (Table 1). Most oviposition sites (63%) were above dense vegetation; 30% of oviposition sites were over moderate vegetation (33 to 66% cover); 6% were over sparse vegetation (1 to 33% cover); and <1% were over unvegetated substrate. Dominant vegetation was sedge-rush (74% of oviposition sites); grasses (23%); moss (1%); and <1% for shrub, cattail, and open soil.

We took data on angle from pond center at 208 oviposition sites in 45 breeding ponds. Angle from pond center had a mean of 317° (95% CI = 291° to 342°), with a plurality (40%) of oviposition sites in the northwestern quadrant (Fig. 2). The mean number of egg masses per oviposition site was 9.7 ($s_{\bar{x}} = 1.5$; median = 3.0; range 1 to 255 egg masses).

Breeding Ponds

We detected breeding in 59 of 104 sites that historically had *R. pretiosa* or that were within the species' movement range of historic or

extant sites. Ponds with breeding were in the Deschutes ($n = 40$), Klamath ($n = 15$), and Willamette ($n = 4$) River basins. The median number of egg masses per occupied breeding pond was 10 (range 1 to 576). Thirty-six of the 59 (61%) occupied breeding ponds had <20 egg masses (Fig. 3). Occupied breeding ponds in the Klamath River Basin averaged 12.4 egg masses ($s_{\bar{x}} = 4.4$; median = 6.5; range 1 to 70; $n = 16$). We found an average of 78 egg masses ($s_{\bar{x}} = 20$; median = 18; range 1 to 576; $n = 43$) in occupied Deschutes/Willamette breeding ponds.

Our sample sites were 32 natural ponds-lakes (2 in Klamath, 30 in Deschutes and Willamette River basins); 17 marshes (4, 13); 24 oxbows (0, 24); 8 beaver ponds (7, 1); and 23 constructed sites (7, 16). We found *R. pretiosa* breeding in 16 natural ponds-lakes (50% of those sampled); 9 marshes (53%); 11 oxbows (46%); 6 beaver ponds (75%); and 17 constructed sites (74%). Average egg mass counts by site type (including unoccupied ponds) were: natural ponds-lakes = 67 ($s_{\bar{x}} = 24$); marshes = 38 ($s_{\bar{x}} = 23$); oxbows = 4 ($s_{\bar{x}} = 1$);

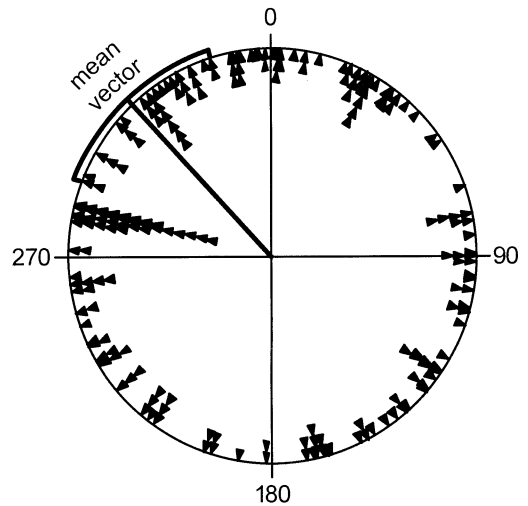


FIGURE 2. Angles from pond centers for *Rana pretiosa* oviposition sites ($n = 208$). Mean vector (μ) is 317° (95% CI: 291°–342°).

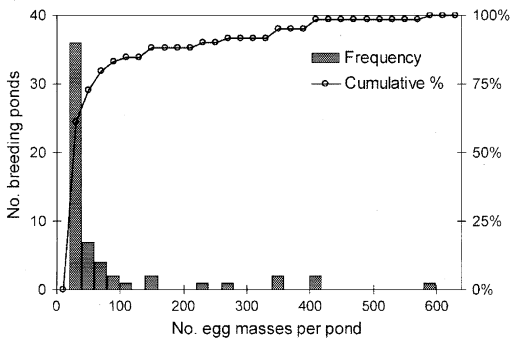


FIGURE 3. Counts and cumulative percentage of *Rana pretiosa* egg masses per occupied breeding pond ($n = 59$) in Oregon.

beaver ponds = $5(s_{\bar{x}} = 2)$; and constructed sites = $35(s_{\bar{x}} = 17)$ (Fig. 4). We found breeding at 1 of the 6 sampled historic localities (Gold Lake) identified by Hayes (1994).

We found introduced American Bullfrogs breeding at only 2 sites (1 each in the Klamath and Deschutes River basins) and both sites also had breeding *R. pretiosa*. We documented Reed Canary Grass at 14 potential breeding ponds, of which 12 had *R. pretiosa* breeding (one of which was Big Marsh). We estimated that 10 sites where Reed Canary Grass was detected had $\leq 3\%$ coverage.

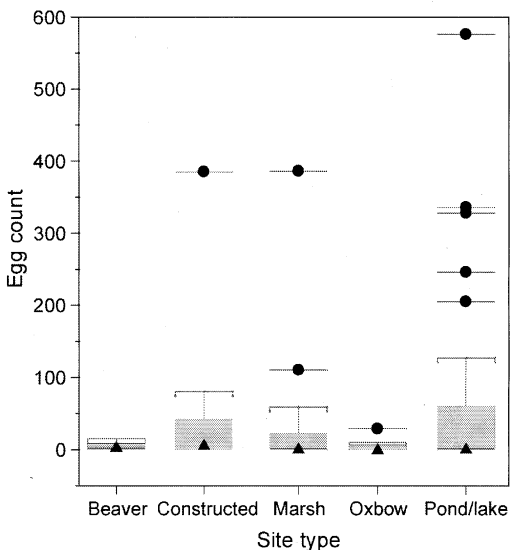


FIGURE 4. Counts of *Rana pretiosa* egg masses per site type in Oregon. Boxes are 25th and 75th percentiles; triangles are medians. Outlying values beyond 1 standard span of 75th percentiles are individually drawn.

Two of the models describing the number of egg masses in potential breeding ponds accounted for $>90\%$ of the Akaike weight (Table 2). Summed weights across the model set for variables with support were as follows (Table 3): *R. pretiosa* breeding nearby: 0.998; non-native fish access to *R. pretiosa* overwinter habitat: 0.956; emergent/submergent vegetation: 0.626; non-native fish access to *R. pretiosa* breeding habitat: 0.373. None of the other predictors had summed weights >0.02 . Basin had a summed variable weight of 1.0 because it was included in all models. The variables for which we found $>50\%$ probability of being in the best model yielded the following predictions for *R. pretiosa* egg mass counts in Oregon: 1) ponds in the Klamath River Basin had 0.63 times (95% C.I. 0.22–1.83) as many egg masses as ponds in the Deschutes/Willamette River basins; 2) ponds that had other breeding nearby had 23.09 times (95% C.I. 5.66–94.14) as many egg masses as those where nearby *R. pretiosa* breeding was not known (Fig. 5); 3) ponds where non-native game fish had access to a majority of local *R. pretiosa* overwintering habitats had 0.11 times (95% C.I. 0.02–0.71) as many egg masses as sites where non-native fish had limited or no access to *R. pretiosa* overwintering habitat; 4) for each doubling in the area of emergent and submergent vegetation in a breeding pond, the number of *R. pretiosa* egg masses increased by a factor of 1.27 (95% C.I. 1.05–1.53) (Fig. 5).

DISCUSSION

Despite the conservation concern for *R. pretiosa*, few data are available to assess broad status of the species and potential influences of habitat and stressors on population size. There are several sites that support large breeding numbers in the *R. pretiosa* core range, but many had fewer than 20 egg masses. We found that the number of egg masses at a pond was associated with the presence of other nearby *R. pretiosa* breeding sites (positive) and with non-native fish access to potential overwintering habitat (negative) more than with the other site characteristics we examined. Pond isolation has been shown to negatively affect population persistence and recolonization of local extinctions in several other ranid frogs (Sjogren 1991; Witte and others 2008). Inter-pond movements by *R. pretiosa* are incompletely understood, but

TABLE 2. Five best-ranked regression models for numbers of *Rana pretiosa* (OSF) egg masses in potential breeding ponds ($n = 104$; Big Marsh not included). Null model is provided for comparison. The Akaike weight (ω) is the likelihood that the model is the best model among the considered group. Basin is included in all models except Null.

Model	# parameters	AICc	Δ AICc	ω weight
OSF Breed Nearby + Fish & OSF Overwintering Hab + Emgt & Submt Veg	5	656.99	0.000	0.581
OSF Breed Nearby + Fish & OSF Overwintering Hab + Fish & OSF Breeding Hab	6	657.91	0.930	0.365
OSF Breed Nearby + Emgt & Submt Veg + Depth	6	663.93	6.945	0.018
OSF Breed Nearby + Emgt & Submt Veg + Fish & OSF Breeding Hab	6	665.61	8.621	0.008
OSF Breed Nearby + Emgt & Submt Veg + Overwintering Hab	5	666.06	9.078	0.006
Null	1	703.40	46.415	0.000

saturated or inundated areas appear to be important for habitat connectivity (Watson and others 2003). One telemetry study from western Washington detected only 1 movement outside the wetland by adult *R. pretiosa* over a period of 3 y (Watson and others 2003). Water >1 cm deep was present at 645 of 654 points (about 99%) identified as frog locations in that study (Watson and others 2003). Minimal movement by *R. pretiosa* outside of flooded or saturated substrate appears to contrast with the behavior of the closely related Columbia Spotted Frog (Bull and Hayes 2002; Pilliod and others 2002). The pronounced effect of proximal sites on numbers of egg masses, along with limited

dispersal capability of *R. pretiosa*, suggest that more isolated ponds that cease to be used for breeding are unlikely to be recolonized except under specific habitat or moisture conditions.

Our findings imply that the effects of non-native fish are mediated by habitat complexity and the seasonal use of microhabitats by *R. pretiosa*. We found substantial evidence for a negative effect of non-native fish with access to spring and channel habitats thought to be favored by *R. pretiosa* for overwintering. This trend suggests that *R. pretiosa* in our study area can benefit from fish-free overwintering sites even if fish are present in other local habitats. We found less support for an effect of non-

TABLE 3. Variable weights, coefficients, and odds ratios for predictors of *Rana pretiosa* (OSF) egg mass counts in potential breeding ponds ($n = 104$; Big Marsh not included). Weights represent the probability that a variable should be in the best model, given the data. The coefficients on categorical variables refer to the estimated effect relative to the lowest-ordered category (see Table 1).

Effect	Variable weight	Coefficient	$s\bar{x}$	Odds Ratio	95% Limits on Odds
Intercept	1.00	-1.74	2.13		
Basin (Klamath) ^a	1.00	-0.45	0.54	0.63	0.22, 1.83
Emgt & Submt Vegetation ^b	0.63	0.34	0.14	1.41	1.07, 1.85
Depth, Level 1	0.02	0.03	0.03	1.03	0.97, 1.08
Depth, Level 2	0.02	0.02	0.03	1.02	0.97, 1.08
Habitat Type, Anthropogenic	<0.01	<0.01	<0.01	1.00	1.00, 1.00
Habitat Type, Beaver	<0.01	<0.01	<0.01	1.00	1.00, 1.00
Habitat Type, Oxbow	<0.01	<0.01	<0.01	1.00	1.00, 1.00
Habitat Type, Pond & Lake	<0.01	<0.01	<0.01	1.00	1.00, 1.00
Grazing, Level 1	<0.01	<0.01	<0.01	1.00	0.99, 1.01
Grazing, Level 2	<0.01	<0.01	<0.01	1.00	0.99, 1.00
Beaver Activity, Level 1	<0.01	0.02	0.02	1.02	0.98, 1.06
Beaver Activity, Level 2	<0.01	0.01	0.01	1.01	0.99, 1.04
Overwintering Habitat	0.01	<0.01	<0.01	1.00	1.00, 1.00
Fish & OSF Breeding Habitat, Level 1	0.37	0.95	0.62	2.58	0.76, 8.79
Fish & OSF Breeding Habitat, Level 2	0.37	0.22	0.35	1.24	0.63, 2.44
Fish & OSF Overwintering Habitat	0.96	-2.17	0.93	0.11	0.02, 0.71
OSF Breed Nearby	1.00	3.14	0.72	23.09	5.66, 94.14

^a Variable weight is 1 because Basin was included in all models.

^b Variable weight and coefficient refer to the ln-transformed variable for wetland vegetation coverage.

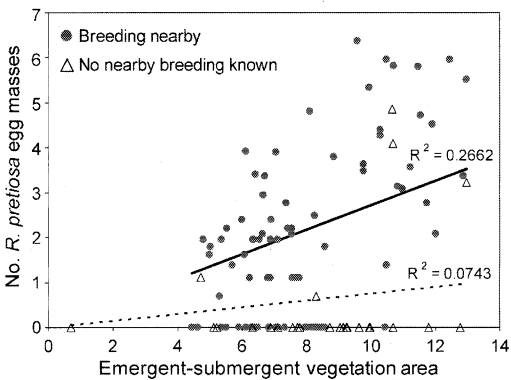


FIGURE 5. Relationship between number of *Rana pretiosa* egg masses per potential breeding pond and amount of Emergent-Submergent vegetation (both \ln -transformed). Grouping variable is whether *Rana pretiosa* are known to breed in nearby ponds (within 0.5 km overland or 2 km by hydrological connection) ($n_{No} = 25$; $n_{Yes} = 79$).

native fish access to oviposition habitats. A stronger negative effect of fish in *R. pretiosa* overwintering habitat than in breeding habitat is consistent with 3 other documented patterns: 1) predation by fish near *R. pretiosa* breeding areas may be reduced by moderate and dense vegetation that is typical of those areas (Baber and Babbitt 2004); 2) predation may be more pronounced in spatially constrained overwintering habitats where frogs and fish may both seek flow and dissolved oxygen (for example, Pilliod and Peterson 2001; Bull and Hayes 2002); and 3) demographic modeling suggests the survival of post-metamorphic stages that overwinter is generally more important than larval survival in affecting population trends (Biek and others 2002).

Similar to reports from sites in Washington State and British Columbia (Licht 1971; Watson and others 2003), *R. pretiosa* in our study area concentrate their egg masses in open, shallow-sloped benches with moderate to dense emergent herbaceous vegetation of grass, sedge, and sedge allies. Factors such as hydrological alterations, invasion by plants that form monocultures, succession toward woody vegetation, and intensive livestock grazing all occur within *R. pretiosa* core range and have potential to affect these microhabitats favored for oviposition. The importance of shallow vegetated habitats was also evident at the pond scale, where we found that egg mass numbers were positively related to the

amount of emergent and submergent vegetation coverage. Shallows with these vegetation types provide feeding areas, refuge from predators, and warmer water, all of which have been posited as benefitting *R. pretiosa* (Licht 1971, 1986; Hayes 1997; Watson and others 2003; Pearl and others 2005).

We found *R. pretiosa* breeding in a broader range of pond types than we expected for a species considered a marsh specialist. Pond type did not enter any of our top models predicting breeding abundance; the high outlier egg mass counts were found in larger sites with greater wetland vegetation, and this is consistent with coverage of preferred vegetation types having a greater influence than pond origin (Fig. 4). These data seem to suggest that *R. pretiosa* can use a variety of pond types as long as they have sufficient vegetation and seasonal habitat available for breeding, summer feeding, and overwintering. We found breeding in 17 anthropogenic ponds, so pond construction may represent a viable option for conserving some threatened populations. Most of the anthropogenic ponds in our study (17 of 23) were in complexes that had other breeding sites nearby. Constructed ponds (average emergent-submergent vegetation coverage 18,474 m²) were generally larger with more wetland vegetation than oxbows (1090 m²) and beaver ponds (2258 m²), but had less vegetation than natural ponds-lakes (43,904 m²) or marshes (91,243 m²). We did not include in our analysis the translocated population at Dilman Meadow, where *R. pretiosa* now breeds in 7 constructed ponds (Chelgren and others 2008). Recently excavated ponds at Dilman Meadow appear to be less attractive for *R. pretiosa* breeding than older ponds that have wetland vegetation and more shallows (Chelgren and others 2008; Pearl and others, unpubl. data). Conservation efforts will benefit from a better understanding of attributes of constructed ponds (including their landscape context) that affect their long-term suitability for *R. pretiosa*.

This study points out areas where further work can improve understanding of *R. pretiosa* status and aid in management for the species. Several lines of evidence suggest that *R. pretiosa* in the Klamath River Basin may be at greater risk than those in the upper Deschutes River system: 1) we found lower egg mass numbers at

Klamath River Basin breeding ponds; 2) there are greater distances and more modified lands between known sites in the Klamath River Basin; and 3) among *R. pretiosa* sites in Oregon that have monitoring data >5 y, the only one with a clear declining trend is in the Klamath River Basin (Jack Creek). Less of the potentially suitable habitat in the Klamath River Basin portion of our study area is public land, so it is possible other occupied sites have gone undetected. An improved understanding of historic and current *R. pretiosa* population trends would help prioritize and implement restoration projects in the Klamath River Basin.

Movement data are needed to better understand relationships between breeding ponds and populations, and their implications for *R. pretiosa* conservation on a landscape scale. Our analysis necessarily focused on the scale of breeding ponds, but this scale does not always equate with a population, and thus may not address factors operating at larger spatial scales. As an illustration, we have some confidence (based on topography and what we have observed of movements) in assigning 37 of our occupied ponds into 21 distinct populations between which interchange of frogs is limited. Our rough estimate is that the 23 other ponds comprise 7 to 13 additional populations, depending mainly on the level of connectivity along riparian zones. We need an improved understanding of how landscape features affect movement and genetic interchange in order to assess the value of riparian corridors and other linkages between sites.

Due to low detections and coverage, we did not assess effects of American Bullfrogs and Reed Canary Grass, 2 invasive species thought to negatively affect *R. pretiosa* in other portions of its range. American Bullfrogs have potential to affect *R. pretiosa* via predation or competition (Pearl and others 2004; Monello and others 2006). Reed Canary Grass monocultures alter oviposition habitat and may affect *R. pretiosa* movements (Watson and others 2003). Abundance of both invasives is currently limited in our study area, but both merit monitoring. American Bullfrogs and Reed Canary Grass are more prevalent in regions outside the current core range where *R. pretiosa* is presumed extirpated. Our lack of detections of both these invasives suggests there is an opportunity to

manage against their expansion into the extant range of *R. pretiosa* in Oregon.

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